Soil carbon changes and uncertainties with New Zealand land-use change

Kevin Tate[1], R.H. Wilde[1], David J. Giltrap[2], W.T. Baisden[1], Surinder Saggar[1], Noel A. Trustrum[1] and N.A. Scott[3]

[1]Landcare Research, Palmerston North, New Zealand. www.landcareresearch.co.nz Email tatek@landcarereasearch.co.nz
[2]Small Office Systems Ltd, Lower Hutt, New Zealand. Email sos@actrix.co.nz
[3]Woods Hole Research Center. Massachusetts, USA. Email nscott@whrc.org

Abstract
An IPCC-based Carbon Monitoring System (CMS) is being implemented to monitor soil C stocks and flows for New Zealand. Geo-referenced soil C data from 1153 sites (0.3 m depth) are used in the current version to assign steady-state soil C stocks to various combinations of soil class, climate, and land use. Overall, CMS estimates of soil C stock are consistent with detailed, stratified soil C measurements at specific sites and over larger regions. Soil C changes accompanying land-use changes were quantified using a national set of land-use effects (LUEs) derived using a General Linear Model. Predicted and measured soil C changes for the grazing–forestry conversion agreed closely. Most soil C is stored in grazing lands (1480±60 Tg to 0.3 m depth), which appear to be at or near steady state; their conversion to exotic forests and shrubland contributed most to the predicted national soil C loss of 0.7±0.3 Tg C yr^{-1} during 1990–2000. Major uncertainties arise from estimates of changes in the areas involved, the assumption that soil C is at steady state for all land-cover types, and from lack of soil C data for some LUEs. Other uncertainties in our current soil CMS include: spatially integrated annual changes in soil C for the major land-use changes; lack of soil-C-change estimates below 0.3 m; C losses from erosion; and the contribution of agricultural management of organic soils.

Key Words
Soil carbon, land-use change, monitoring, New Zealand

Introduction
Land-use change has featured prominently in the development of the New Zealand landscape (26.7 Mha) over the past millennium. Polynesians, followed by European settlers, cleared over half the area of indigenous forests, with vegetation carbon (C) losses of 3.4 Pg. Since the mid-1800s, pastures have been established for sheep and cattle grazing, probably increasing mineral soil C slightly. Conversion from sheep to dairy farming on flat to gently rolling grazing land has also recently occurred. The propensity of some of these soils to erode, however, could have resulted in large, but highly uncertain, soil C losses. Over the last 2 decades, many hectares of pasture have been converted to plantation forests (primarily *Pinus radiata* D.Don – current area of 1.7 M ha), or reverted to indigenous woody vegetation (shrublands). In both cases, C can accumulate quickly in the vegetation, but the implications for soil C changes are unclear because the techniques to quantify current and future C stocks and changes in soils at large spatial scales are poorly developed. The impact of various land-use practices on soil C stocks has been recently reviewed (Guo and Gifford 2002), including large-scale assessments associated with the impacts of agricultural management (e.g., Lal *et al*. 1998). In 1996, the New Zealand Ministry for the Environment commissioned the development of a soil Carbon Monitoring System (CMS) to report soil CO_2 emissions in response to land-use changes under the UN Framework Convention on Climate Change. New Zealand has now ratified the Kyoto Protocol, and to achieve its CO_2-emissions-reduction target and gain C credits must account for C stocks and changes above- and below- ground in its Kyoto forests (SRLUCF 2000). Accordingly, we have developed a national monitoring system for soil to quantify baseline C stocks (Scott *et al*. 2002), and changes associated with present and future land-use changes (Tate *et al*. 2003a,b).

The two key land-use changes in New Zealand during the past decade or so have been conversion of grazing land to managed exotic forests, and reversion to natural vegetation. Although stocks and harvest-removals in managed forests are accurately quantified (Forest Research 2001), soil C changes in these forests (Tate *et al*. 2003a,b) and in all revegetating land are difficult to estimate because changes may be small. Cropland represents a major land-use change globally, but the contribution of soil C changes from this land use to New Zealand's national C budget is very small. This is due to the negligible area under field crops and horticulture (1.2% of the total land area) and the small rate of conversion. The effect on soil C of recent adoption of forage cropping by dairy farmers may need to be factored in to this assessment in the near future.

We here first describe the recent development and testing of the soil CMS to reduce the uncertainties in New Zealand's soil C stocks, and changes resulting from the effects of land-use change and management. Then we identify major sources of uncertainty, including annual changes in soil C with key land-use changes, the effects of erosion on national estimates of soil C changes, and management effects on organic soils.

Approach

National C stock changes

The vegetation CMS measures changes in C stored in indigenous forest and shrublands, based on a network of permanently marked plots on an 8-km^2 grid, to provide an unbiased estimate of C stocks and changes for mainland New Zealand and the near offshore islands (Coomes *et al*. 2002). To date, over 300 plots have been established. When complete in 2007, the plot network will contain 1400 plots. Remeasurement will likely occur at 5- or 10-year intervals.

We have designed a National Soil CMS to quantify effects of land-cover change on soil C storage in New Zealand (Scott *et al*. 2002) (Figure 1). Our system is similar to the default system proposed by IPCC, but uses national soil pedon data to derive country-specific estimates of soil C for different soil, climate, topography and land-cover classes. The system assumes changes in land cover or land use are the key drivers of annual and decadal changes in soil C, and relies on periodic updating of land cover to estimate these changes. Retrospective land-cover-change comparisons are possible for some land uses (e.g., managed forests), but for others (e.g., revegetating indigenous scrub), area changes are generally poorly quantified. An equilibrium soil C value has been ascribed to various combinations of land-cover type, soil type, climate and topography (slope x rainfall). The topography factor is strongly correlated with the propensity of a site to erode and is a useful predictor of soil C. The geo-referenced soil C and bulk density dataset comprises 1178 sites with soil C data to 0.1 m, 1153 with soil C to 0.3 m, and 886 with soil C to 1 m. The dataset includes 55, 49 and 16 sites to 0.1, 0.3 and 1 m depth, respectively, for exotic forest. These data were analysed against these combined predictors using a Generalized Linear Model. The observed soil C data were best explained by the additive model: soil-climate + land-cover + slope x rainfall, giving a single land-use effect (factor) across all 18 soil-climate categories. This indicated data for all soil-climate and land-use combinations were not needed to calculate soil C changes with land-use change and that estimates of change (with uncertainties) in soil C can reasonably be ascribed to causal effects of land use rather than chance land-use selection effects. Land-use effects (LUE; Mg C ha^{-1} ±SEM) for key land-cover types (0–0.3 m mineral soil C), compared with improved pasture (LUE, 0), were: arable –11±8, horticulture –9±7, managed forests –16±7, indigenous forest –1±5, scrub –12±5. Changes in land-cover area since 1990 were obtained or inferred from national agricultural statistics since 1990 and from the land- cover database (MAF 2001).

Testing

We have undertaken limited testing of the CMS, including its ability to predict a) soil C stocks at regional (e.g., Scott *et al*. 2002) and local (Tate *et al.* 2003a) scales, and (b) soil C changes on converting improved ryegrass/white clover pastures to *P. radiata* D. Don plantations (Tate *et al*. 2003b). These changes are compared with a number of estimates using paired-sites (Davis and Condron 2002; Halliday *et al*. 2003).

Only limited data exist yet for the grassland-to-shrubland comparison, but the effects of management changes on soil C in grasslands (14.0 M ha) were tested using a range of published and unpublished site studies (Saggar *et al*. 2001).

Results and discussion

New Zealand's soil C stocks

Nationally, soil C stocks to three depths (0–0.1, 0.1–0.3, and 0.3–1 m), estimated by the additive model, were 1300±20, 1590±30, and 1750±70 Tg. These estimates were higher than earlier estimates (Scott *et al*. 2002) for 0–0.1 m (12%), 0.1–0.3 m (11%) and 0.3–1 m depth (9%). Database revision and recent sampling of categories occupying large land areas were mainly responsible for these changes. Inclusion of topography (slope x rainfall) in the current model corrects for over-estimation of the land-use effect on soil C for some combinations of land-cover types, and is a surrogate for erosion effects on soil C. Regional-scale (24 000 ha) testing of the earlier version (Scott *et al.* 2002) of the soil CMS showed that including both soil and climate produced a more accurate estimate of soil C stocks than did soil type alone. The CMS also predicted soil C accurately for 15 out of 19 combinations of soil-climate and land use in an area of 6000 ha where all major (grazing land, exotic and indigenous forest, shrubland) and some minor (arable) land-cover types were represented. We also compared soil C values in the CMS database with values obtained from random field

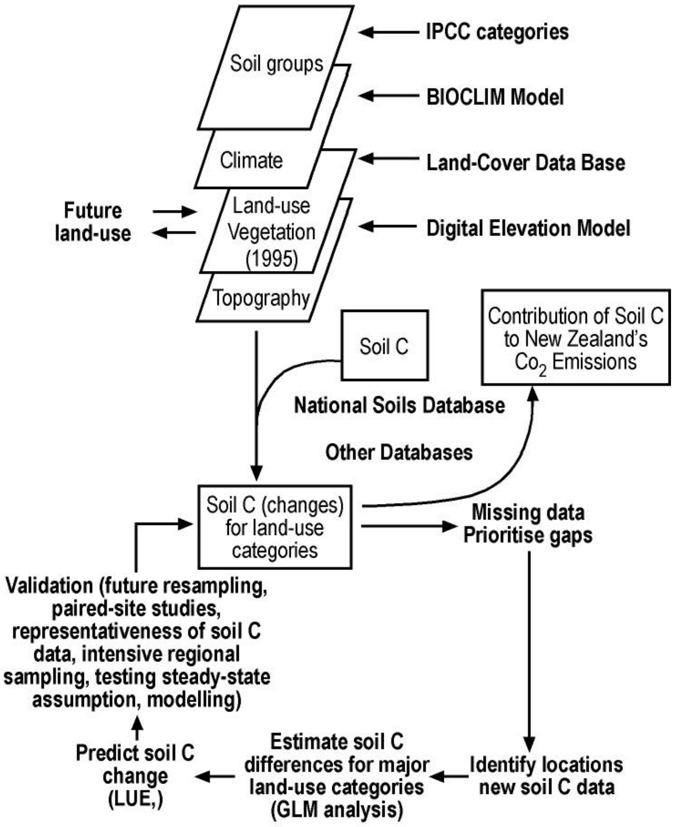

Figure 1. The structure, development, testing and use of the soil Carbon Monitoring System.

sampling for Temperate Volcanic soils under pasture. Mean soil C data from random sampling were well within 95% confidence limits of the database values, indicating that (a) soil C from historical samples are representative of current soil C concentrations, and (b) there have been no discernable land management effects since the database soil C data were collected.

Remnant indigenous forests (6.25 M ha, 1140±90 Tg vegetation C) and shrubland (ca. 2.7 Mha, 190 Tg vegetation C) also contain much of New Zealand's soil C. The largest store of soil C in New Zealand, however, is found in grazing land (1480±60 Tg C), occupying 14 Mha. Most contemporary land-use changes occur from these grasslands, principally pastures, so the potential for C loss from these soils is high.

Changes in soil C due to land-use and management changes 1990–1999.

For the current analysis, grazing land includes both improved (mainly fertilised, legume-based pastures) and unimproved grassland (including tussock). Soil C at all depths for these two grassland classes is similar. We
used improved grassland as the reference against which to compare soil C changes because (a) almost all contemporary land-use changes include improved grassland and (b) a number of published and unpublished reports (Saggar *et al.* 2001) suggest soil C for established grassland can be assumed to be at steady state. Accordingly, we have set the change estimate for land-management effects on grassland at zero (Table1).

Since 1990, more than 80% of new managed forests (mainly *P. radiata)* have been planted on improved and unimproved pastures. Current national reporting of C accumulation in these forests includes C in the forest floor, but not in the mineral soil. We have examined changes in mineral soil C using the soil CMS and data from paired-site studies where the forest ages exceeded 10 years (Davis and Condron 2002; Halliday *et al*. 2003). Linear changes in soil C were assumed for this comparison.

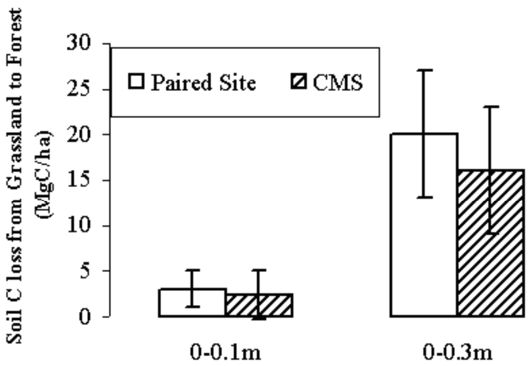

Figure 2. Soil C differences determined by the soil CMS and paired- site studies. Differences are from grazing land to exotic forest. Error bars indicate one standard error.

The LUE for exotic forest (0–0.3 m soil depth) compared with grazing land from the CMS (–16 ±7 Mg/ha) is close to, but less precise than, the corresponding estimate (–20 ± 7 Mg/ha) from the paired-site data (Figure 2). However, as our predicted value is based on a much larger dataset than the paired-site estimate, we used the former to estimate that about 0.7 ± 0.3 Tg C yr^{-1} could have been lost from mineral soil between 1990 and 2000, as a result of afforestation of grazing land (Table 1). While this loss is small compared with the C gain in the vegetation, it probably offsets any C increase in the forest floor. The need to understand historical land-use effects and ecosystem C cycling over several rotations points to a need for process-based models. Measured soil C changes associated with the transition from grassland to exotic forest can be simulated at specific sites using process-based models of soil C turnover (Halliday *et al*. 2003; Scott *et al*. 2004) (Figure 3).

These simulations using ROTHC show that from forest establishment to canopy closure (years 1–10) soil C losses are greatest, whereas from about year 10 to harvest (ca 26 years) losses progressively decline (Scott *et al*. 2004) These soil C changes can largely be explained by, initially, losses of readily decomposed soil C from the grassland being much greater than the C inputs from the young trees. Then, as the more readily decomposable grassland C declines and C inputs from the growing trees continue to increase, the net loss of soil C becomes progressively lower.

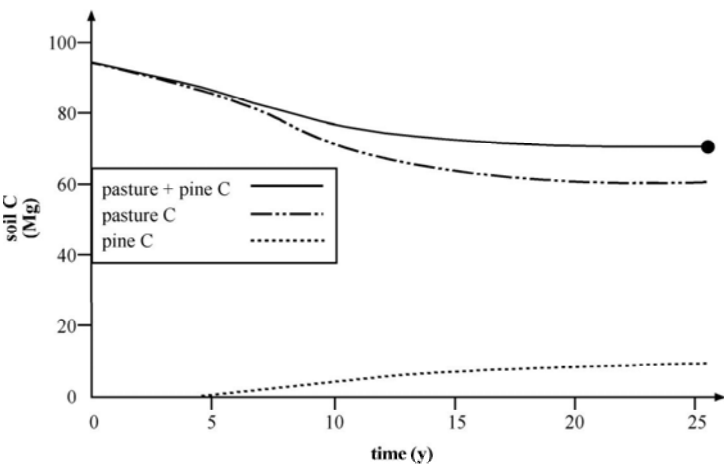

Figure 3. Soil C changes (0–0.3 m depth) with afforestation of improved pasture during one rotation, Tikitere, North Island (adapted from Scott *et al*. 2004). Measured data at years 0 (not shown) and 26 (●).

Despite progress in realistically simulating these soil C changes at specific sites, our present understanding of key processes and parameters limits our ability to use such models in a predictive sense at larger scales. Key processes include the decomposition of surface litter, translocation of C and N from forest floor to mineral soil, and erosion.

Since 1985, much former grazing land has reverted to mainly indigenous shrubland, and exotic species (e.g., *Ulex europaeus, Cytisus scoparius*). Poor statistics exist for the areas undergoing this land-use change, and current estimates are inferred from changes in pasture and indigenous forest areas. According to the soil CMS, some soil C loss may occur when shrublands regenerate on abandoned grazing land, but the few litter C estimates available suggest litter C may offset mineral soil C loss (Tate *et al*. 2003a). These predicted soil C losses are consistent with some paired-site studies, but only at the wetter end of rainfall gradients.
Some studies exist of soil C changes with conversion to and from pastures for some grain crops, but not for horticultural conversion which appears to have increased recently (ca 3000 ha/y).

Apart from the use of a topography factor (slope x rainfall) as a surrogate for erosion, no account is taken in this analysis of the effects on the terrestrial C budget of soil erosion, which are potentially much larger than direct land-use change effects but currently very uncertain (Tate *et al.* 2000).

Soil erosion
Land-use and land-cover changes have featured prominently in the development of the New Zealand landscape. Before human settlement, primary forest covered about 23 M ha (85%) of New Zealand's 26.7 M ha of land area. After Polynesian settlement, the primary forest area was reduced gradually to about 13.5 M ha as a result of burning and clearing. European settlement led to an additional rapid loss of just over 7 M ha. Overall, C losses from deforestation could total 3.4 Pg C. Much of this forest area was converted to pasture for sheep and cattle grazing.

Accompanying this latter phase of deforestation has been a marked increase in soil loss by erosion (e.g., East Cape, North Island), exacerbated by a combination of high tectonic activity, youthful landscapes and a climatic regime that is periodically disturbed by high-intensity rainstorms. In addition to reducing the productive capacity of the land, soil erosion is also responsible for moving very large amounts of C (in the form of organic matter and plant residues) from New Zealand to the ocean each year. We are developing a methodology using physically based models to estimate the erosion, transport, deposition and decomposition of soil C at landscape to national scales.

Table 1. Some New Zealand soil C (0–0.3 m) stocks in 1990, and recent average changes with land- use change.

Land use	Soil C stocks (1990) and changes (1990–2000)			
	Area (Mha)	Soil C (Tg)	Areal change (kha y^{-1})	Change in Soil C (Tg y^{-1})
Grazing land	14.0	1480 ± 58	– 48	0
Exotic forest	1.3	77 ± 23	+ 52	– 0.7 ± 0.3[a]
Natural (shrub) vegetation	2.7	244 ± 18	–8	0
Cropland	0.3	26 ± 3	+ 3	– 0.03 ± 0.02

[a] Calculated assuming 85% forest planted in grazing land.

Currently, estimates of C loss, based on intersecting improved national-sediment-delivery ratios and soil C estimates, and total riverine export of particulate and dissolved organic C, are about 3±1 Tg C y^{-1}, at the lower end of the range from 3–11 Tg C y^{-1} previously reported (Tate *et al*. 2000). This estimated soil C loss equates roughly with the amount of C sequestered annually during the period 1990–2000 by New Zealand's Kyoto forests.

Organic soils
Because it estimates soil C stocks and changes within a given depth range, our system excludes organic soils, which cover ca 0.07% of New Zealand. An assessment of changes in CO_2 emissions from changes in organic soil requires organic C mass changes to be estimated. Drainage of organic soils for agricultural or forestry use generally results in rapid and sustained subsidence of the peat surface, due to a combination of changes in physical conditions and microbial decomposition. In the Waikato region of New Zealand, as much as 110 000 ha of peat have been drained and a large area converted to pasture.

Published on CDROM. Website www.regional.org.au/au/asssi/

Schipper and Mcleod (2002) suggested average subsidence rates of 3.4 cm y^{-1} during 40 years of dairy farming may be contributing as much as 0.5 Tg CO_2- C to New Zealand's annual emissions. A more recent study (Joost Nieven, unpublished results) of CO_2 exchange from drained Waikato peat soils under pasture suggests these emissions eventually subside to quite low levels. Further research is now needed to reduce this uncertainty in the contribution of drained peat soils to New Zealand's national C budget.

Conclusions

Based on the key land-use changes occurring in New Zealand over the past few years, associated changes in soil C are not likely to exceed ca 1 Tg C/y. There are still large uncertainties associated with this estimate, which current paired-site, process-modelling, and erosion studies are seeking to address.

Acknowledgements

The contribution of a number of Landcare Research and Forest Research scientists to the initial development of the current version of the soil CMS is gratefully acknowledged. This work is supported by the Ministry for the Environment and the New Zealand Foundation for Research, Science and Technology.

References

Coomes DA, Allen RB, Scott NA, Goulding C, Beets P (2002) Designing systems to monitor carbon stocks in forests and shrublands. *Forest Ecology and Management* **164,** 89–108.

Davis MR, Condron L (2002) Impact of grassland afforestation on soil carbon in New Zealand: a review of paired-site studies. *Australian Journal of Soil Research* **40,** 675–690.

Forest Research (2001) Carbon inventory of New Zealand's planted forests. (Forest Research: Rotorua, New Zealand)

Guo, LB, Gifford RM (2002) Soil carbon stocks and land-use change: a meta analysis. *Global Change Biology* **8**,345–360.

Halliday JC, Tate KR, McMurtrie RE, Scott NA (2003) Mechanisms for changes in soil C storage with pasture to *Pinus radiata* land-use change. *Global Change Biology* **9**, 1294–1308.

Lal R, Kimble J, Cole CV, Follett R. (1998) In '*The potential of US Cropland to Sequester Carbon and Mitigate the Greenhouse Effect'.* (Ann Arbor Press: Ann Arbor, Michigan)

MAF (2001) New Zealand Ministry of Agriculture and Forestry Statistics Website www.maf.govt.nz/statistics/primaryindustries/landcover/index.htm

Saggar S, Tate KR, Hedley C, Perrott K, Loganathan, P (2001) Are soil carbon levels in our established pastures at or near steady state? *NZ Soil News* **49 (4)**, 73–78.

Schipper LA, Mcleod M (2002) Subsidence rates and carbon loss in peat soils following conversion to pasture in the Waikato region, New Zealand. *Soil Use and Management* **18,** 91–93.

Scott NA, Tate KR, Giltrap DJ, Smith CT, Wilde RH, Newsome PF, Davis MR (2002) Monitoring land-use change effects on soil carbon in New Zealand: quantifying baseline soil carbon stocks. *Environmental. Pollution* **116,** S167–S186.

Scott NA, Tate KR, Ross D.J, Parshotam A (2004) Processes influencing soil carbon storage following afforestation with *Pinus radiata* on a New Zealand Andisol. *Australian Journal of Soil Research,* In Review.

SRLUCF (2000) *Land Use, Land-Use Change and Forestry*. A Special Report of the IPCC. (Eds RT Watson, IR Noble, B Bolin, NH Ravindranath, DJVerardo, DJDokken). (Cambridge University Press: Cambridge).

Tate KR Scott NA, Parshotam A, Brown L, Wilde RH, Giltrap DJ, Trustrum NA, Gomez B, Ross DJ (2000) A multi-scale analysis of a terrestrial carbon budget. Is New Zealand a source or sink of carbon? *Agriculture, Ecosystems and Environment* **82**, 229–246.

Tate, KR, Scott N.A, Saggar S, Giltrap D.J, Baisden WT, Newsome PF, Trotter CM, Wilde RH (2003a) Land-use change alters New Zealand's terrestrial carbon budget: Uncertainties associated with estimates of soil carbon change between 1990 and 2000. *Tellus* **55B**, 364–377.

Tate K.R, Barton JP, Trustrum NA, Baisden WT, Saggar S, Wilde RH, Giltrap D.J, Scott NA (2003b) Monitoring and modeling soil organic carbon stocks and flows in New Zealand. In 'Soil Organic Carbon and Agriculture: Developing Indicators for Policy Analysis'. (Ed. CAScott-Smith).

Proceedings of an OECD Expert Meeting, Ottawa, Canada, October 2002. (Agriculture and Agri-Food Canada and Organisation for Economic Co-operation and Development: Paris) 329 p.